Nature Conservation 19: 19–37 (2017)
doi: 10.3897/natureconservation.19.12457
http://natureconservation.pensoft.net

RESEARCH ARTICLE

A peer-reviewed open-access journal
Nature Conservation
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Patterns of distribution and landscape connectivity of the stag beetle in a human-dominated landscape

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Academic editor: *G.M.Carpaneto* | Received 28 February 2017 | Accepted 12 June 2017 | Published 31 July 2017

http://zoobank.org/AF08AEC0-BB47-402D-AE1A-39298E6DFA92

Citation: Della Rocca F, Bogliani G, Milanesi P (2017) Patterns of distribution and landscape connectivity of the stag beetle in a human-dominated landscape. In: Campanaro A, Hardersen S, Sabbatini Peverieri G, Carpaneto GM (Eds) Monitoring of saproxylic beetles and other insects protected in the European Union. Nature Conservation 19: 19–37. https://doi.org/10.3897/natureconservation.19.12457

Abstract

Urbanisation and the spread of agriculture have resulted in high levels of forest loss, habitat fragmentation and degradation in many regions of the world. In Italy, the Po Plain is the most human-dominated landscape of the country and, after decades of exploitation, old-growth forests have been reduced to small and isolated patches, often threatened by invasive tree species such as the black locust (*Robinia pseudoacacia*). In these habitats, the occurrence of many forest-dependent species is related to the quality and availability of suitable areas, as well as the connectivity between the remaining forested patches.

Thus, recently developed species distribution models have been applied, namely the Ensemble of Small Models (ESMs), to identify areas of occurrence for a rare and protected saproxylic beetle species, the stag beetle *Lucanus cervus* and the inverse of the resulting distribution maps as resistance maps have been used to estimate landscape connectivity for this species.

Response curves suggested that the probability of the stag beetle occurrence increased with habitat diversity, grassland coverage and native forests, especially oak and mixed forests. The other forest coverage, such as those with black locust, beech, chestnut and black cherry, showed a unimodal relationship peaking approximately at 70%, 8%, 55% and 13% respectively. The stag beetle occurrence was unimodal related to distance to watercourses and distance to human settlements and negatively related to shrub-lands, croplands, sparse and dense human settlements. Landscape connectivity showed similar patterns, except for oak forest coverage, which showed a negative relationship to landscape connectivity.

In conclusion, stag beetles can persist in a human dominated landscape only in the presence of forest patches, including those with black locust trees. It is also inferred that ESMs may be suitable for modelling rare species distributions and estimating landscape connectivity to promote species conservation.

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Keywords
Circuitscape, *invasive species*, Lucanus cervus, Robinia pseudoacacia, *Species Distribution Models*

Introduction

Urbanisation and intensive agriculture have resulted in high levels of forest loss and natural landscape fragmentation in many areas of the world (Foley et al. 2005). In Italy, new human settlements (buildings, industrial and commercial areas, as well as infrastructure) take over 70 hectares of natural or agricultural land per day (Munafò and Tombolini 2014). The remaining natural areas are not only embedded in a highly disturbed matrix of human-transformed land cover, but often serve as a continuous source for non-native species and become population "sinks" that are unable to support self-sustaining populations of the native species (McKinney 2002).

In this context, many forest-dependent animals are at risk of extinction, especially those with limited dispersal abilities, such as many saproxylic insects. Usually, these species are highly specialised and are linked to specific forest resources such as the amount of dead wood and particular conditions of temperature, humidity and fungal associations (Ranius and Fahrig 2006, Stokland et al. 2014). Amongst forest species affected by habitat fragmentation, the stag beetle *Lucanus cervus* (Linnaeus, 1758) is one of the flagship species adopted for saproxylic insect guilds (Thomaes et al. 2008). Indeed, even though the stag beetle is able to disperse well, reaching about 700m with a single flight (Rink and Sinsch 2007), it is dependent on sites with high spatio-temporal continuity of habitat related to natural forests and it is therefore used as an indicator species for habitat continuity (Müller et al. 2015).

Thus, forest planning and management should explicitly include connectivity assessments, identifying the most suitable forest sites for the maintenance of habitat connectivity in human dominated landscapes (Pascual-Hortal and Saura 2007).

Species distribution models (SDMs) have helped conservation planning for threatened species by identifying sites in which environmental conditions are favourable, especially in those areas where the species is not present and where restoration programmes could therefore be focused (Guisan et al. 2013). However, their predictive accuracy decreases drastically due to model overfitting when species with limited occurrences (such as rare species) and multiple predictor variables are considered in the models (Lomba et al. 2010, Breiner et al. 2015). This problem has recently been overcome by a new set of models called "Ensemble of Small Models" (ESMs), which are able to limit model-overfitting for rare species and provide more accurate predictions compared to standard SDMs (Breiner et al. 2015).

Landscape connectivity is increasingly estimated through circuit theory-based methods (McRae et al. 2008) which conceptualise the landscape as a conductive surface within an electrical circuit. Species locations or favourable habitat patches often represent the nodes of the network and the connections between nodes are determined by underlying resistance surfaces, based on ecological assumptions about the move-

ments of the species within the landscape (Tournant et al. 2013). Due to limited information regarding factors affecting animal movement and dispersal, SDMs are increasingly and successfully used to develop resistance surfaces (Wang et al. 2008, Milanesi et al. 2016a, b) and attention recently shifted from ecological corridor identification amongst species locations to independent node-based models (Koen et al. 2014).

Although recent studies developed conservation frameworks using ESMs to identify high conservation priority areas (Breiner et al. 2015, D'Amen et al. 2015, Benito et al. 2016) and independent node-based models to estimate landscape connectivity (Koen et al. 2014, Pitman et al. 2016), to the authors' knowledge, a combination of ESMs and independent node-based models for threatened species conservation in human-dominated landscapes have never been carried out to date.

Thus, the aims of this paper were (i) to identify habitat requirements for the stag beetle in a human-dominated landscape by developing ESMs, (ii) to identify ecological corridors for this species using the resulting probability of occurrence map of ESMs as a resistance surface in independent node-based models and (iii) to verify whether the above mentioned invasive tree species are related to stag beetle occurrence and landscape connectivity.

The empirical data set consisted of the locations of the stag beetle from one of the most human-dominated landscapes in Europe, the Po Plain in Italy (Ingegnoli 2015). In fact, after decades of agriculture and industrial activity, the old-growth forests of the Po Plain have been reduced to small and isolated patches (Blondel and Aronson 1999) and the remaining forests are also greatly threatened by human-introduced invasive tree species such as the black locust (*Robinia pseudoacacia*) and the black cherry (*Prunus serotina*; Annighöferet al. 2015). Even if previous research has also shown saproxylic insect occurrence in habitats affected by the invasive black locust (Della Rocca et al. 2016), further tests on the effect of invasive species on native species occurrence and connectivity are needed.

Materials and methods

Study area

The study was conducted in an area of about 1,025 km^2 within the Varese Province in the Lombardy region of northern Italy. Elevation in the study area ranges from a few metres above sea level (a.s.l.) near the Ticino River to about 600 m a.s.l at the foothills of Campo dei Fiori, above the Varese lake. The climate is temperate sub-continental (Ferré et al. 2005), generally humid with a mean annual temperature of about 13°C and an annual precipitation of between 700mm and 1,200mm (Prigioni 1995). This area encompasses twelve Natura 2000 sites amongst which were included nine Sites of Community Importance (SCI) and three Special Protection Areas (SPA) (Fig. 1; Suppl. material 1: Table S.1). Due to its strategic geographical location, this area connects the most extensive biodiversity source of the Po Plain, represented by the Valle del Ticino, to the massive mountains

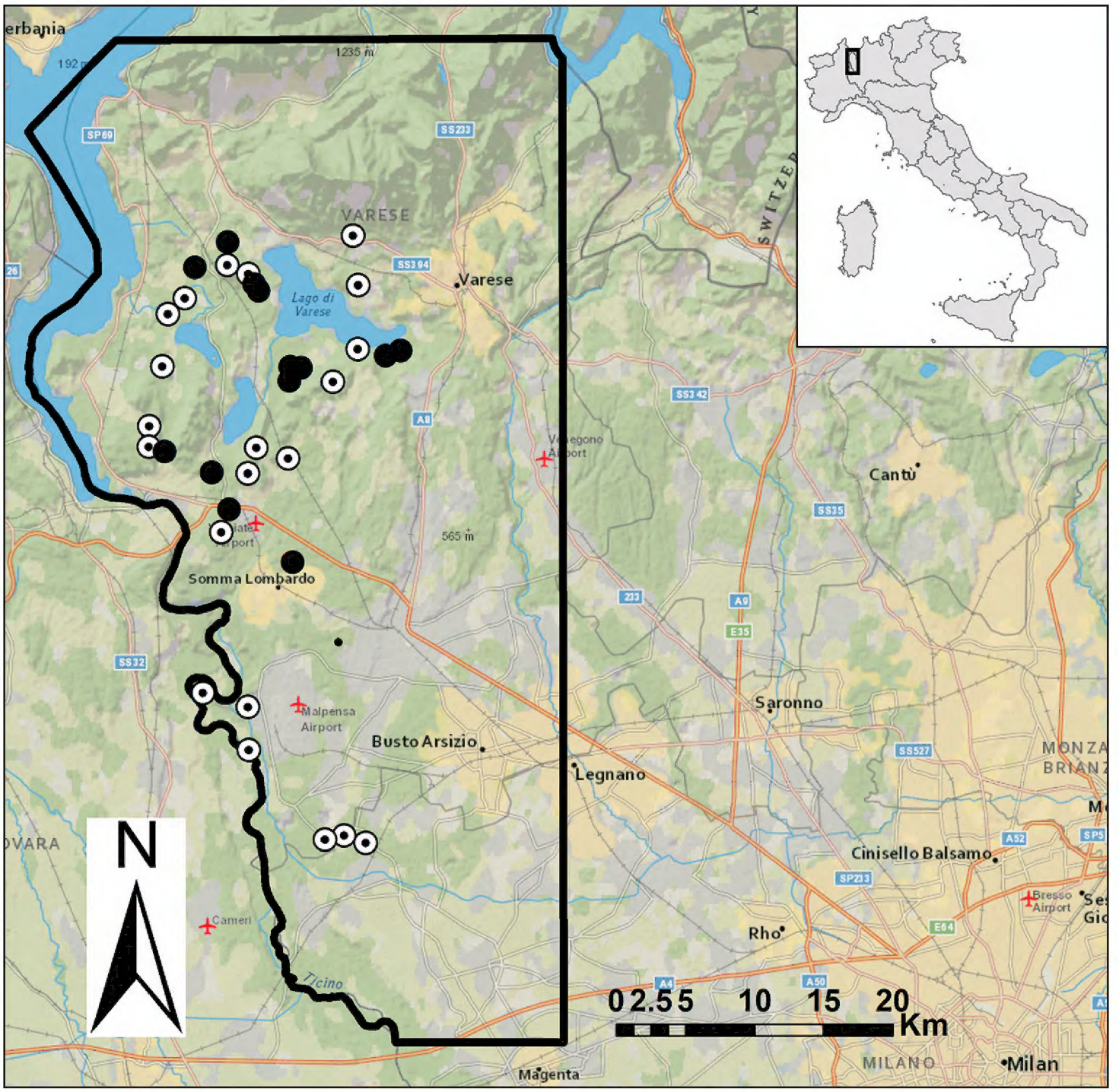

Figure 1. Study area and stag beetle sampling locations (black dots with white circles). Filled black dots indicate investigated sites where the species was not recorded. Black lines indicate the borders of the study area.

of Campo dei Fiori (Casale et al. 2015) and it is an important ecological corridor between the Alps and the Apennines (Casale and Brambilla 2008), the two major "reservoirs of biodiversity" present in the Italian peninsula and included amongst the worldwide Priority Ecoregions for Global Conservation (Olson and Dinerstein 1998). The role of the study area as an ecological corridor is confirmed by the fact that two very ecologically exigent forest species such as the pine marten (*Martes martes*) and the black woodpecker (*Dryocopus martius*), which only lived in mountainous and forest areas until a few years ago, recently began colonising lowland areas of the Po Valley (Casale et al. 2015).

Despite its important ecological role and the high biodiversity value of the Ticino Valley Regional Park, which has been acknowledged as the MAB Biosphere Reserve "Valle del Ticino" (UNESCO 2005), the study area crosses the most urbanised area of the country, including the Milano-Malpensa International Airport. The forests of

this area are very fragile, fragmented and mainly concentrated along the riverbanks. Wherever broad roads and clearings have been opened up, the dominant tree species, including the pedunculated oak (*Quercus robur*), black alder (*Alnus glutinosa*) and hornbeam (*Carpinus betulus*), have been rapidly replaced by a scrub of exotic species consisting of a dense undergrowth of black cherry (*Prunus serotina*), black locust (*Robinia pseudoacacia*), tree of heaven (*Ailanthus altissima*) and red oak (*Quercus rubra*) (Bruno et al. 2008).

Study species and data

The target species, the stag beetle, is considered as threatened in several countries within its geographical range (Harvey et al. 2011), listed in the IUCN Red List of Threatened Species as "near threatened" and in the EU Habitats Directive as a priority species of community interest (Appendix II) (Nieto and Alexander 2010). It is considered a focal species for the conservation of suitable habitats for saproxylic beetles (Thomaes et al. 2008) and as an umbrella species, as it represents a diverse and highly vulnerable fauna associated with a wide range of broad-leaved trees (Percy et al. 2000) and coarse woody material (Buse et al. 2008, Ducasse and Brustel 2008). Therefore, by directing management efforts toward the requirements of this species, the requirements of many cohabitant species that use the same habitat are addressed (Roberge and Angelstam 2004). The dataset consists of locations of stag beetles collected in the context of the TRANS INSUBRIA BIONET (TIB) LIFE project (LIFE10/NAT/IT/241) and the AMBROSIANO Regional project. Both of these projects aimed to identify, improve and protect ecological corridors for endangered species in highly fragmented and human-dominated areas. Species occurrences were investigated from 2012 to 2015 in 22 sites, from 2013 to 2015 in 6 sites and from 2014 to 2015 in 6 more sites for a total of 34 sites within the study area. Each site was surveyed every year, for two months (June and July), once a week. To minimise detection bias due to changes in stag beetle activity throughout the day, each site was surveyed in three different time periods: once a month at sunset and three times per month alternatively in the morning or in the afternoon. Each survey consisted of one transect, 500 m length and 10 m width, walking along roads or paths within wooded areas at a constant speed of 17m/min, for a total duration of 30 minutes in dry and low-windy days (Campanaro et al. 2016). The exact locations (X, Y coordinates) and the gender of all specimens observed flying or walking on the ground within transects were recorded.

Predictor variables

A set of 18 predictor variables were derived that were contiguously available for the entire study area (Table 1). Specifically, the percentage of oak forests, native broad-leaved forests, invasive broad-leaved forests, mixed forests, beech forests, chestnut woodlands,

Table 1. Variables used in the development of stag beetle (*Lucanus cervus*) Ensemble of Small Models. Variables with Variance Inflation Factor (VIF) > 3 have to be removed due to multi-collinearity with other variables. Average values ± standard deviations at sampled and presence sites are also shown.

Variable	VIF	Sampled sites	Presence sites
Native broad-leaved forests (%)	2.854	26.35 (±38.21)	28.76 (±40.71)
Mixed woods (broad-leaved and coniferous) (%)	2.507	19.06 (±38.25)	19.62 (±39.54)
Oak forests (%)	1.081	9.87 (±23.74)	14.66 (±29.31)
Beech woods (%)	1.001	6.34 (±26.93)	8.85 (±34.07)
Chestnuts woods (%)	1.629	3.76 (±17.21)	6.11 (±21.75)
Distance to watercourses (m)	1.416	804.24 (±576.81)	859.34 (±626.99)
Shrub-lands (%)	1.121	7.85 (±6.11)	9.27 (±7.01)
Grasslands (%)	1.843	14.12 (±22.02)	10.48 (±19.12)
Croplands (%)	2.155	10.01 (±21.31)	10.67 (±23.13)
Invasive broad-leaved forests (%)	1.002	3.14 (±2.25)	3.09 (±2.22)
Black cherry woods (%)	1.006	3.12 (±2.41)	3.21 (±2.51)
Black locust woods (%)	1.979	9.41 (±22.71)	8.95 (±23.32)
Other woods (%)	1.011	4.56 (±12.36)	5.50 (±15.16)
Shannon diversity index of habitats (unitless)	1.307	1.81 (±0.59)	1.71 (±0.56)
Dense human settlements (%)	2.961	2.18 (±4.88)	1.91 (±6.15)
Sparse human settlements (%)	2.989	4.59 (±13.47)	4.76 (±14.41)
Distance to human settlements (m)	2.582	166.57 (±130.69)	180.35 (±148.06)
Distance to roads (m)	1.701	221.89 (±257.03)	167.67 (±200.11)

black cherry stands, black locust stands and other woods were derived from the map of forest types in Lombardy 2016 (vector data; http://www.geoportale.regione.lombardia.it/). The percentage of shrub-lands, croplands, grasslands and dense and sparse human settlements were derived from the map of destination of agricultural usage and forest soils of Lombardy 2012 (vector data; http://www.geoportale.regione.lombardia.it/). The Shannon diversity index of habitats was also calculated considering all predictors mentioned above. The distance to watercourses and to human settlements was calculated from the map of destination of agricultural usage and forest soils of Lombardy 2012, while the distance to roads was estimated from the Open Street Map (http://www.openstreetmap.org). All of these predictor variables were re-sampled at 100 × 100m scale resolution to match the average home-range size of the target species (Sprecher-Uebersax 2003; Thomaes et al. 2008).

To avoid multi-collinearity amongst predictors, the Variance Inflation Factor (VIF) was calculated. Following Zuur et al. (2010), predictor variables with VIF values > 3 (highly related to other predictors) should be removed.

Ensemble of small models

The ESMs approach is based on the development of all the possible bivariate models (only two predictors at a time out of a larger set of predictors), followed by their

combination into an ensemble (Lomba et al. 2010, Breiner et al. 2015). ESMs were developed with seven widely used modelling techniques (see Appendix S.1). Similar to Breiner et al. (2015), ESMs were developed using locations where the species was recorded and 10,000 random points were generated to serve as pseudo-absence data (including sampled sites where the species was not recorded).

20-fold split sampling (90% training data and 10% test data) was used to evaluate the bivariate models and the resulting ESMs. Similarly to Breiner et al. (2015), all bivariate models with Somers' D (also known as Gini coefficient; i.e. 2 × AUC-1) values equal to or less than 0 (i.e. AUC ≤ 0.5 and thus worse than a random model) were not included in the ESM. The area under the receiver operating characteristic curve (AUC) and the continuous Boyce index were used to evaluate model performance. ESM outputs in an ensemble prediction (EP) weighted by Somers' D values were averaged and thus the resulting map (ranging between 0 and 100) was used in the further analyses. Finally, considering the cells with values above the cut-off point identified by the EP, areas of potential presence for stag beetles were identified. The R package 'ECOSPAT' (Di Cola et al. 2017) in R version 3.2.4 (R Core Team 2013) was used to develop, evaluate and project ESMs.

Structural connectivity and ESMs

A resistance map was derived as the inverse (1 – probability of occurrence) of the resulting map from ESMs and thus it was combined with circuit theory to explore landscape connectivity (McRae et al. 2008). Circuit theory was chosen over other commonly used connectivity methods as circuit theory models movement ecology via random walk pathways across all available movement possibilities (McRae et al. 2008). The software Circuitscape v.4.0.5 (McRae et al. 2008) was used, implementing circuit theory principles and assuming that the intensity of current flow between two nodes (locations) is proportional to the number of times an individual goes from one node to another moving through the particular cell under consideration (McRae et al. 2008). Thus, current flow can be used as a measure of the probability of movement between the two nodes to provide continuous estimates of landscape connectivity for the species considered (McRae et al. 2008). To estimate current flow, Circuitscape simultaneously considers all possible pathways connecting nodes (locations) accounting for an underlying resistance map to movement, producing a map where current density varies across pixels (analogous to the probability of use by random walkers; Koen et al. 2014).

Since high current is produced near nodes, using species locations as nodes could lead to a biased estimation of landscape connectivity (Koen et al. 2014) and thus, similarly to Pitman et al. 2016, a landscape-scale permeability map was developed independent from the former source or destination locations. Specifically, 10–300 random nodes were placed at intervals of 10 around the perimeter of the study area and then Circuitscape was run to identify the optimum number of random nodes required to generate an unbiased landscape connectivity map (Koen et al. 2014). After each iteration, the Pearson correlation was calculated to compare the resulting current density to

those estimated, placing 1,000 random locations within the study area; a current density map was considered independent of node placement when the curve comparing correlation coefficients to the number of node pairs reached an asymptote (Koen et al. 2014; Pitman et al. 2016). Subsequently, the final current map was rescaled between 0 and 1 (indicating low and high current, respectively).

Results

A total of 222 specimens were found, 167 males and 55 females, in 21 of the 34 sites monitored (Table 2).

Multi-collinearity was not detected amongst the predictors (VIF > 3) and thus they were all considered in further analyses (Table 1). These 18 predictor variables resulted in a total of 153 bivariate predictor combinations. The mean values of AUC and the Boyce index for the ESMs were 0.945 (± 0.007 SE) and 0.903 (± 0.081 SE), respectively (Suppl. material 1: Table S.2). The cut-off value identified by EP was equal to 20 and thus a total of 34.29% of the study area above this threshold was estimated. These areas corresponded to 351.31km^2 divided into 935 patches where the stag beetle could potentially occur (Fig. 2).

The probability of occurrence for the stag beetle increased with habitat diversity and native broad-leaved forests, oak forests and grassland. The other forest coverage, such as those with black locust, beech, chestnut and black cherry showed a unimodal relationship peaking approximately at 70%, 8%, 55% and 13% respectively (Fig.3).

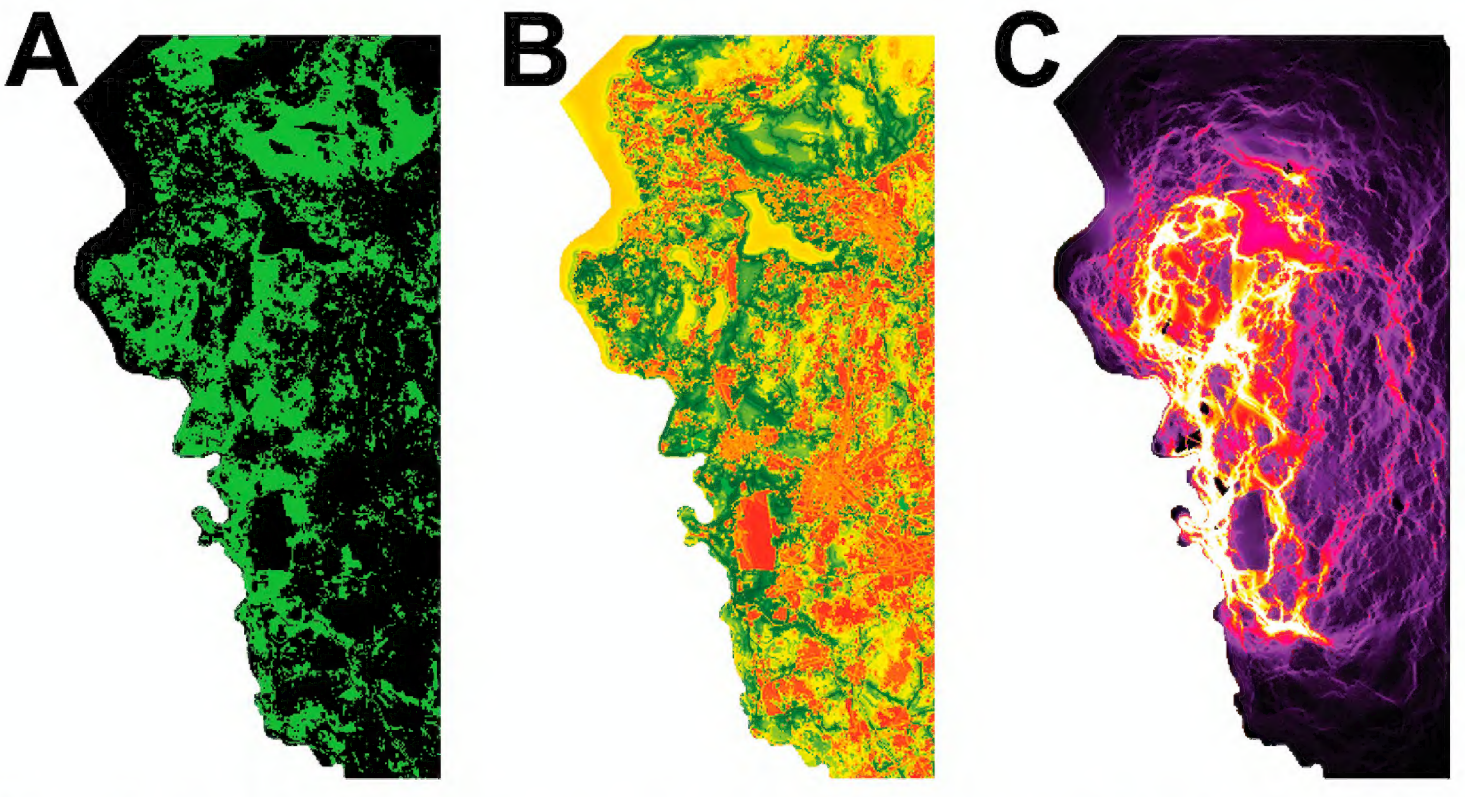

Figure 2. a Suitability areas for the stag beetle derived by the Ensemble of Small Models (green and black areas indicate suitable and unsuitable areas, respectively) **b** Resistance surface for the stag beetle (green-red scale indicates lower-higher resistances, respectively) derived by the Ensemble of Small Models, ESMs (1- probability of occurrence) **c** current map for the stag beetle (black-yellow scale indicates lower-higher connectivity, respectively) derived from resistance surface using Circuitscape software.

Table 2. Number of specimens collected within the study area in each sampling site from 2012 to 2015. (F= Female; M= Male).

Site	Locality	Years								Total	
		2012		2013		2014		2015		M	F
		M	F	M	F	M	F	M	F		
1	Buguggiate	0	0	0	0	0	0	0	0	0	0
2	Azzate	0	0	0	0	0	0	0	0	0	0
3	Galliate lombardo	1	1	2	0	1	0	2	0	6	1
4	Casale litta	0	0	0	0	0	0	0	0	0	0
5	Inarzo	0	0	0	0	0	0	0	0	0	0
6	Inarzo	0	0	0	0	0	0	0	0	0	0
7	Vergiate	5	1	4	2	0	1	2	1	11	5
8	Casciago	0	0	1	0	1	0	1	1	3	1
9	Barasso	1	0	2	0	3	0	3	2	9	2
10	Biandronno	0	0	0	0	0	0	0	0	0	0
11	Biandronno	0	0	0	0	0	0	0	0	0	0
12	Besozzo	0	0	0	0	0	0	0	0	0	0
13	Malgesso	0	1	1	0	1	0	2	0	4	1
14	Brebbia	0	0	0	0	0	0	0	0	0	0
15	Travedona-Monate	2	1	0	1	1	0	3	0	6	2
16	Cadrezzate	1	0	1	0	2	0	1	0	5	0
17	Cadrezzate	10	5	4	2	4	1	4	0	22	8
18	Taino	5	0	5	0	2	0	1	1	13	1
19	Vergiate	0	0	0	0	0	0	0	0	0	0
20	Sesto Calende	0	0	0	0	0	0	0	0	0	0
21	Vergiate	2	0	1	0	1	0	2	0	6	0
22	Bodio Lomnago	1	0	1	0	1	0	1	0	4	0
23	Bregano	–	–	0	2	1	0	3	0	4	2
24	Vergiate	–	–	1	0	1	0	1	0	3	0
25	Vergiate	–	–	2	0	1	0	2	1	5	1
26	Arsago Seprio	–	–	0	0	0	0	0	0	0	0
27	Sesto Calende	–	–	0	0	0	0	0	0	0	0
28	Sesto Calende	–	–	2	0	1	0	1	0	4	0
29	Castano Primo	–	–	–	–	11	8	7	3	18	11
30	Castano Primo	–	–	–	–	2	0	3	1	5	1
31	Lonate Pozzolo	–	–	–	–	2	0	4	1	6	1
32	Vizzola ticino	–	–	–	–	7	4	5	3	12	7
33	Vizzola ticino	–	–	–	–	6	3	3	5	9	8
34	Vizzola ticino	–	–	–	–	8	2	4	1	12	3

The probability of occurrence for the stag beetle was unimodal related to distance to watercourses and distance to human settlements (with a peak at 800m and 180m respectively) and negatively related to shrub-lands, croplands, sparse and dense human settlements and increased distance to roads (Fig. 3).

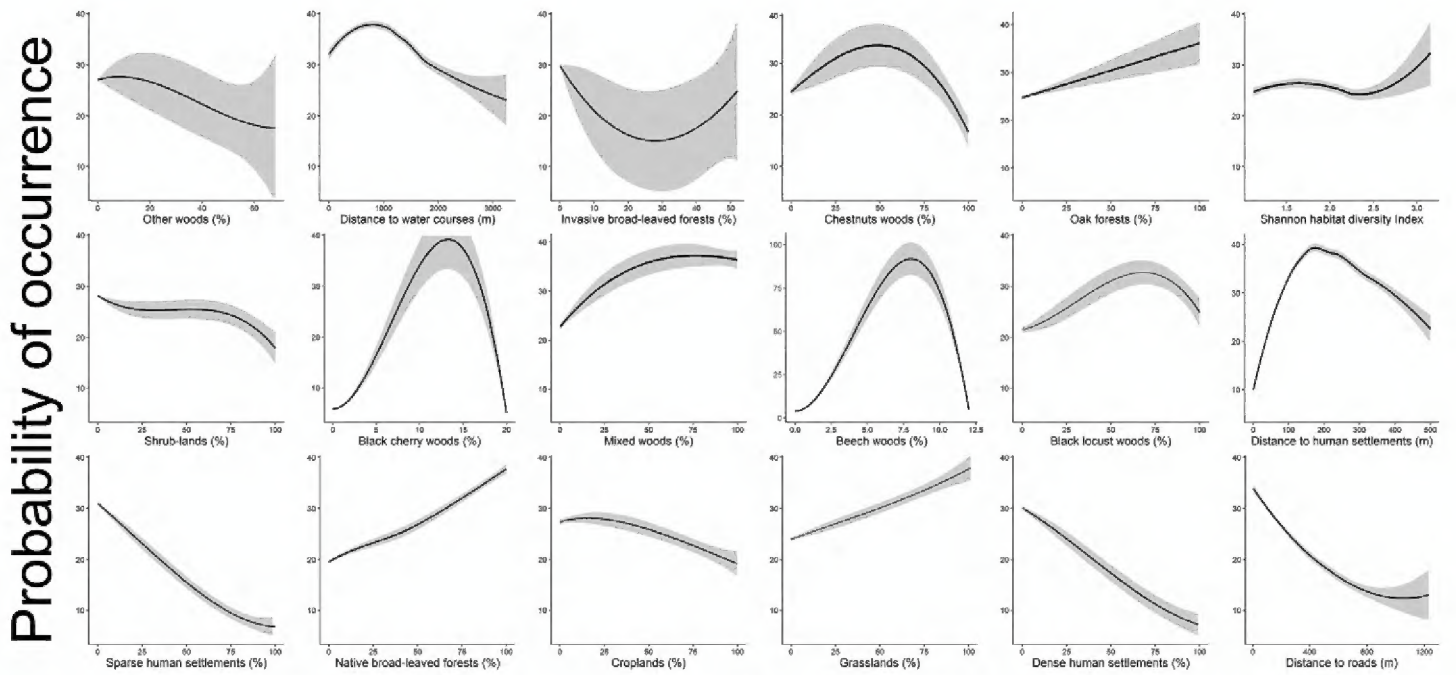

Figure 3. Response curves and 95% confidence intervals (in grey) of the probability of occurrence of the stag beetle derived by the Ensemble of Small Models in relation to predictor variables values.

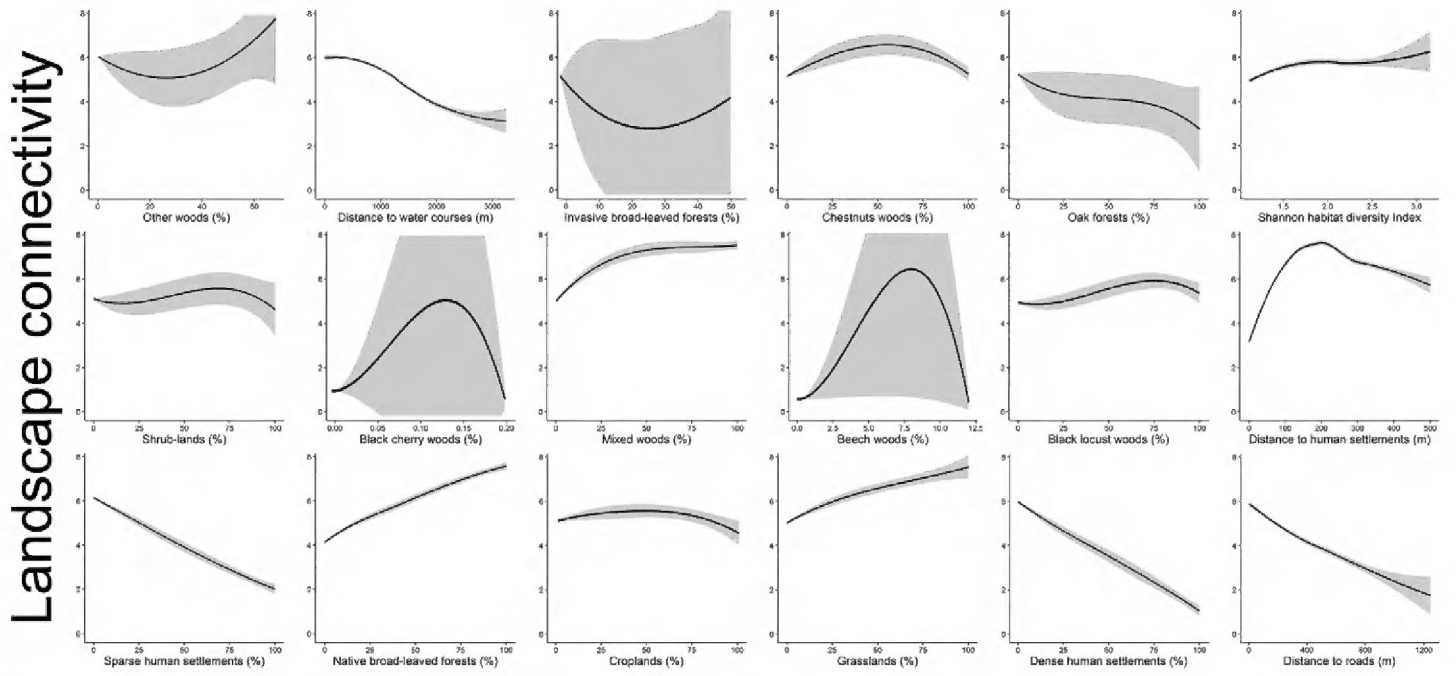

Figure 4. Response curves and 95% confidence intervals (in grey) of landscape connectivity of the stag beetle estimated through Circuitscape in relation to predictor variables values.

Using the inverse of the probability of occurrence as a resistance surface (Fig. 2), an unbiased landscape connectivity map was developed using ≥ 120 random nodes. In fact, the curve comparing Pearson correlation coefficients to the number of node pairs reached an asymptote at the value of 120 node pairs (Suppl. material 1: Fig. S.1) and thus the current density map was considered independent of the node placement (Fig. 2).

The landscape connectivity showed similar patterns of relationships with the considered predictor variables, except for oak forest coverage which tended to decrease as landscape connectivity increased (Fig. 4).

Discussion

This study took place within projects aimed at preserving the natural habitats in a human-dominated landscape that play a crucial role in connecting the Mediterranean basin to Northern Europe. In this context, suitable areas were identified for stag beetle reproduction and fundamental corridors for this species during dispersal. The most recent and robust species distribution and landscape connectivity modelling techniques were applied. The results confirmed that occurrence and connectivity of the beetles is related to natural habitats instead of anthropogenic habitats. Moreover, the research highlighted a complex puzzle in how managing invasive tree species (such as black locust thicket) may provisionally help in maintaining native animal populations in human-dominated landscapes. The research also identified that ESMs may be suitable for modelling rare species distributions and estimating landscape connectivity, provided that detectability problems are overcome.

Ensemble of small models and landscape connectivity

SDMs and landscape connectivity based on resistance surfaces are not without caveats. On one hand, the predictive accuracy of SDMs decreases considerably when rare species are considered in the models, as few species' occurrences and many predictor variables lead to model overfitting and thus reduced generalisation and applicability of the models (Lomba et al. 2010, Breiner et al. 2015). On the other hand, empirical knowledge about the factors facilitating (or hindering) individuals from moving across landscapes is often limited (Graf et al. 2007) and thus resistance surfaces are often derived from expert opinions. These can be potentially biased by the experience of the expert(s), scarce knowledge about the study area and limited or lacking information and published literature on the ecology of species (Zeller et al. 2012, Stevenson-Holt et al. 2014).

However, ESMs can overcome model-overfitting for rare species and thus provide more accurate predictions compared to standard SDMs (Breiner et al.2015) and it has been shown that ensemble predictions perform better compared to single modelling techniques (Marmion et al. 2009). Moreover, SDMs are increasingly and successfully used to derive resistance surfaces, avoiding subjective expert opinions (Wang et al. 2008, Milanesi et al. 2016a, b) and, recently, attention has shifted from single ecological corridor identification between species locations to independent node-based models (Koen et al. 2014).

Despite their high predictive accuracy, ESMs have not been often used to model rare species distribution and they have never been used to derive resistance surfaces to model landscape connectivity. Thus, in this study, a novel application of ESMs has been provided and it is inferred that they might be valuable tools for estimating unbiased landscape connectivity. Since the presence of rare species and landscape connectivity are amongst the most frequently cited criteria for site selection by conservationists

(Tuomisto 2010), by combining these two important aspects of biodiversity, ecologists are strongly encouraged to implement this framework in order to develop successful management policies and conservation actions.

Patterns of distribution and landscape connectivity of the stag beetle

Suitable habitats for the stag beetle in the study area were mostly located in the largest patch of contiguous forest along rivers and lakes within the two natural parks, the Ticino Valley Regional Parkand Campo dei Fiori Park. Less suitable habitats were identified in the south-eastern part of the study area, which includes an intensive agricultural matrix, the Milano Malpensa airport and several cities surrounding Milan.

The relationship between the probability of occurrence predicted by ESMs and the predictor variables considered in this study is consistent with the ecological requirements of the species (Thomaes et al. 2008, Harvey et al. 2011). Specifically, response curves showed that forests and grasslands were the most suitable habitats for the stag beetles, while shrub-lands, croplands and human settlements (both sparse and dense) were largely avoided. On a landscape level, woodlands were the primary habitat type for this saproxylic species, since they are the main sources of high amounts of coarse woody debris, (Ranius and Kindvall 2006, Davies et al. 2008). In fact, in many European countries, the stag beetle is mainly associated with oak woodlands and, secondarily, with other tree species such as beech, chestnut and willow (Harvey et al. 2011). As expected, oak and mixed forests were found with the highest probability of occurrence followed by chestnut forests. Some invasive woodland, such as black locust and lack cherry, also seemed to be suitable for the species. In particular, the black locust forests ensured the maximum probability of occurrence for the stag beetle if they extended to about 70% of the landscape. However, over this threshold the stag beetle occurrence decreased rapidly.

This result is of interest as no previous information on the use of invasive woodlands by the stag beetles has been available.

The idea of a negative impact of invasive tree species on native species and ecosystems is generally supported (Maerz et al. 2005; Powell et al. 2011). However, most of these studies refer to herbaceous and shrub invasive plants, while the effect of the widespread invasive tree species on animal communities is still hotly debated. When considering saproxylic beetles, a general avoidance for some exotic coniferous tree species was recorded for *Pinus brutia* in Israel (Buseet al. 2010) and for *Pseudotsuga menziesii* in Germany (Müller et al. 2015). A recent study, however, carried out in the temperate forests of northern Italy (Della Rocca et al. 2016), demonstrated that the black locust dead wood, as well as that with oak, hosts a wide range of saproxylic beetles and thus, acts as a surrogate for native deadwood for saproxylic beetles. Some studies supported the hypothesis that the use of exotic plants by native insects increases with extension and time since their introduction (Brändleet al. 2008; Branco et al. 2015). Thus, considering the wide distribution of the black locust and its presence over a long period

in the north Italian forests, it is likely that the stag beetle could use this exotic species similarly to other native species, such as oaks.

It was found that the black locust forest can contribute to stag beetle occurrence if its coverage does not exceed 70% of the landscape. This means that the remaining 30% should be represented by other tree species or forest types, especially oak trees and broad-leaved forests. It is possible that, despite the extensive spread of invasive forests, the occurrence of the stag beetle is assured by suitable natives tree species that remain in a small proportion scattered in the invaded area. This occurred in northern Spain, where the stag beetle was found in a Eucalyptus plantation due to the presence of very old chestnut trees remaining within those plantations (Marco Mendez *personal comm*).

These results showed that the stag beetle remains within the proximity of urban settlements and is positively affected by the presence of roads although it does not seem to be as anthropophilic as in other European countries such as Belgium or Great Britain (Thomaes et al. 2008, Harvey et al. 2011). As suggested by Thomaes et al. (2008), the thermophilous character of the species (Napier 2003) could drive it to the warmer microclimate of cities. Thus, since the Italian mean summer temperatures are considerably higher than those of central and northern Europe (5th assessment of the Intergovernmental Panel for Climate Change, IPCC AR5WG1 2013; http://www.ipcc.ch), the species remains distant from urban centres and only uses open human infrastructure, such as roads, to facilitate flight and to warm themselves before flight (Harvey et al. 2011).

Considering landscape connectivity, the stag beetle was directly related to the high coverage of woodlands. However, oak forests did not seem to be as important for the species during movement and dispersion compared to other forest types. This phenomenon is probably due to the difference that naturally exists between the habitats used for dispersion and those used for reproduction, as has already been reported for other species (Blàzquez-Cabrera et al. 2016). Actually, while the role of oaks as primary resources for the reproduction of the stag beetle is widely recognised (Thomaes et al. 2008), it is less critical for its dispersion.

Finally, human settlements (both sparse and dense) were the main barriers for movement and dispersion of the stag beetle, while open habitat, especially grasslands and roads, represented important components for the species movement.

Management implications

A successful management policy for the protection of threatened animal species in human-dominated landscapes should take into account strategies for ensuring the persistence of good-quality habitats and landscape connectivity (Rudnick et al. 2012). In fact, for those forest-dependent species which barely survive in fragmented landscapes dominated by human settlements and intensive agriculture, it is fundamental to plan precise and effective management actions for ensuring suitable breeding sites and minimising costs and efforts (Laycock et al. 2009). This is the case for stag beetles living in the highly urbanised Po Plain of northern Italy.

According to a recent report on the distribution and conservation status of species and habitats of Community Interest in Italy (Genovesiet al. 2014), for almost all saproxylic species, including the stag beetle, the main pressure factors are represented by forestry and silvicultural practices, such as removal of dead wood and stumps for economic and civic purposes and felling old-growth trees in both forests and urban parks. Due to the development of ecological corridors for the stag beetles in this study area, sites were identified along the corridors to maintain ancient woods, forest remnants, hedgerows and old deciduous trees, represented by both native and exotic species. In between this network of strict conservation areas, tree retention and islands of leftover parts of trees and snags after harvest (Heikkala et al. 2014; Vandekerkhove et al. 2013) can be implemented. In this way, larval habitat, represented by deadwood on the ground or rotting stumps, is provided and the survival of stag beetle populations is assured (Van der Sluijs et al. 2004).

These findings also showed that black locust deadwood can be considered in forest management operations aimed at restoring habitats for the reproduction of the stag beetles and which can serve as a temporary food source for the larvae. However, in light of a conservation strategy for this saproxylic beetle, it is essential to ensure the presence of other forest types, especially mixed broadleaved forests and to use black locust only in combination with oak deadwood.

Acknowledgments

This study was supported by the project LIFE 10/NAT/IT/241 – TIB_TRANS INSUBRIA BIONET "Connessione e miglioramento di habitat lungo il corridoio ecologico insubrico Alpi – Valle del Ticino and the project AMBROSIANO "Studio per il miglioramento della connettività ecologica tra il Parco Alto Milanese e il Parco del Ticino: contenimento dell'ambrosia ed incremento della biodiversità". We thank Guido Bernini, Silvia Stefanelli and the students Alessantro Tonali and Riccardo Resente for helping during data collection and to Giuseppe Maria Carpaneto, Alessandro Campanaro, Marco Mendez and an anonymous reviewer for their helpful comments on a former version of this manuscript.

Special issue published with the contribution of the LIFE financial instrument of the European Union.

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Supplementary material I

Additional information

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Data type: species data

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Link: https://doi.org/10.3897/natureconservation.19.12457.suppl1